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# Sediment macrofauna communities at a small mussel farm in the northern Baltic proper

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The use of blue mussel farming in the Baltic Sea for mitigating eutrophication through nutrient bioextraction has recently been under debate, with emphasis on potential local negative effects induced by farmed mussels, based on theoretical scenarios. Here, we broaden the discussion by presenting an assessment of the faunal community around a non-commercial mussel farm, after two years of operation in a relatively well-oxygenated water area. It is the first large-sized farm (25 tonnes) in the northern Baltic proper. The biotic conditions at this mussel farm did not deviate negatively from the surrounding reference areas. The total abundance and species richness of sediment-dwelling fauna were higher at the farm site, which also showed a distinct zoobenthic species composition including many bioturbators which may provide extra protection against hypoxia. A snapshot view of water quality indicated relatively low P and chlorophyll-*a* concentrations at the farm. We caution against drawing conclusions from our study that are too far-reaching primarily because the mussel farm in question being smaller than those referred to in the debate. However, the study presents an important initial assessment of the functioning of a pilot mussel farm in the Baltic proper and provides baselines for future investigations and guidelines for improved sampling design.

## Introduction

The utility of blue mussel (*Mytilus trossulus*) farming as a nutrient reduction measure for the Baltic Sea has recently been debated revealing an increased awareness of its positive and negative effects. Stadmark and Conley (2011, 2012) highlighted the risks of waste products from the mussels leading to hypoxic and even anoxic events beneath the farms, which in addition could increase the release of ammonium and phosphorus from sediments and reduce nitrogen release from sediment denitrification processes. They were also concerned about the concentration of organic material from large areas beneath a farm and the effects this could have on biogeochemical processes. Rose *et al*. (2012) replied to these concerns arguing that, by means of a

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simple mass balance calculation, the harvest of shellfish biomass must result in a net nutrient removal, although site-specific and regional differences may constitute a problem when evaluating the full functionality of mussel farming for nutrient bioextraction. Petersen *et al*. (2012) listed several additional advantages of mussel farming as a mitigation tool emphasizing its simplicity as an *in situ* net-remover of nutrients in eutrophic areas; its capability of recycling nutrients of diffuse origin back from sea to land and its positive effects on water transparency thereby potentially stimulating photosynthetic macrophyte (and oxygen) production at greater depths. Petersen *et al*. (2012) also argued, based on their calculations on Danish mussel farm data, that the negative biogeochemical processes beneath the farms are small compared with the basin-scale reduction in sedimentation rates and the removal of nutrients through harvesting. Calculations based on mussel farms on the Swedish west coast (Carlsson *et al*. 2012) support the latter argument by Petersen *et al*. (2012). The contrasting views in this debate converge on the need for thorough information regarding sediment nutrient processes before mussel farms are widely used in nutrient reduction programs.

Furthermore, the urgent lack of real time operational data from a mussel farm in the Baltic proper, an area of special concern, appears as one of the most apparent gaps in knowledge hitherto. Globally, however, there is a steady growing literature on the environmental impact of mussel and shellfish farming (Mattson and Lindén 1983, Kaspar *et al*. 1985, Kaiser *et al*. 1998, Stenton-Dozey *et al*. 1999, Mirto *et al*. 2000, Chamberlain *et al*. 2001, Christensen *et al*. 2003, Crawford *et al*. 2003, Fabi *et al*. 2009, Ysebaert *et al*. 2009, McKindsey *et al*. 2011, 2012, Wong and O'Shea 2011, Ivanov *et al*. 2013, Wilding and Nickell 2013, Neofitou *et al*. 2014). Since filter-feeding mussels do not require external feed input (Dumbauld *et al*. 2009), mussel aquaculture is generally considered to have little negative environmental impact and always less than fish farming (Fabi *et al*. 2009, McKindsey *et al*. 2012). Nonetheless, biodeposits and shells accumulated under or around the farm are rich in both C and N (Kautsky and Evans 1987) and have caused changes in the benthos. These changes/effects range from positive stimulatory (e.g. Wong and O'Shea 2011, Wilding and Nickell 2013, Neofitou *et al*. 2014), via undetectable (e.g. Chamberlain *et al*. 2001, Crawford *et al*. 2003, Fabi *et al*. 2009, McKindsey *et al*. 2012), to locally harmful (e.g. Stenton-Dozey *et al*. 1999, Mirto *et al*. 2000, Chamberlain *et al*. 2001, Christensen *et al*. 2003, Ysebaert *et al*. 2009) and disastrous (e.g. Mattsson and Lindén 1983, Kaspar *et al*. 1985, Ivanov *et al*. 2013). Mild organic enrichment frequently causes an increase in macrobenthic abundance and diversity due to increased food supply (Pearson and Rosenberg 1978, Kraufvelin *et al*. 2006a, Diaz *et al*. 2012). Ongoing nutrient loading consumes oxygen and enriches sulphides in the sediment, thereby gradually changing the composition of macrobenthic communities (Weston 1990). If the nutrient load is heavy, the sediments may even become azoic (Pearson and Rosenberg 1978, Diaz and Rosenberg 1995). Due to factors such as the absence of mussel farms in the Baltic Sea area hitherto, still very little data exist on the effects of mussel farming on macrobenthic communities, although Kotta *et al*. (2009a) showed that mussels may stimulate benthic production locally. With regard to the impact of and recovery from fish farming, however, some Baltic Sea studies are available (Bonsdorff *et al*. 1997, Kraufvelin *et al*. 2001, Villnäs *et al*. 2011), but in most of these cases, nutrient loads were very high (many tonnes) and persistent (> 10 years).

The central issue of the mussel farm debate seems to be to demonstrate and determine the conditions under which farming effects occur and the extent of the influence. It is indeed impossible to negate the accumulation of sediments beneath mussel farms (Hartstein and Stevens 2005, Carlsson *et al*. 2009), although there is high variability in the effects of organic sediment accumulation, which is primarily due to the location of the mussel farm (hydrologic conditions); the net production in tonnes of mussels, the salinity and the farming methods used (Dahlbäck and Gunnarsson 1981, Miron *et al*. 2005). Although effects of increased sedimentation on benthic biota have not been thoroughly discussed, accumulation of organic material is positively correlated with the oxygen consumption and ultimately negative effects can thus be deter-

mined by assessing the biota (Baudinet *et al*. 1990, Nizzoli *et al*. 2011). Still, the benthic ecosystem is dynamic and sediment communities comprise a diverse fauna, including bioturbators, which play a strong role in mixing and maintaining well-aerated sediments. Thereby, the zoobenthos may function as a buffer against hypoxia and anoxia (Norkko and Shumway 2011, Norkko *et al*. 2012). For example, common Baltic bioturbators such as annelid worms *Hediste diversicolor* and *Marenzelleria* spp., as well as the bivalves *Macoma balthica*, *Cerastoderma glaucum* and *Mya arenaria*, can consume fecal pellets directly and remove organic matter from the sediment surface, thereby reducing the risk of oxygen depletion (Norkko and Shumway 2011). Additionally, blue mussels are ecological engineers that facilitate the recruitment of several invertebrates including bioturbators (Westerbom *et al*. 2002, Koivisto *et al*. 2011), which can be transported from mussel farms to the sediment through mussel clumps accidentally falling off farm ropes. Therefore, the abundance and species composition of invertebrates in sediments beneath mussel farms should always be compared with adequate reference areas as an essential part of the determination of mussel aquaculture impacts.

To shed some light on environmental issues related to mussel farming in the Baltic Sea, we examined the conditions at a mussel farm in Kumlinge (Åland Islands) in the Finnish Archipelago Sea. This mussel farm functioned between June 2010 and November 2012, obtaining a harvestable production of 25 tonnes, and has to date been the only large facility operating in the Baltic proper over a relatively long time (Lindahl 2012, Engman 2013). The mussel farm was situated at the same site as a former fish farm (operational until 2008). Blue mussels grew there at the edge of their distribution range (at salinities of ca. 6 psu), presenting slow mussel growth, small mussel sizes and low final biomass (Kautsky 1982, Westerbom *et al*. 2002). The environmental conditions at this mussel farm can thus be considered unusual and different to those outside the Baltic Sea, such as many of the cases referred to in the mussel farm debate. Towards the end of August 2012, a few months before mussel harvesting, we recorded sediment organic levels and macrofauna communities. Additionally, we took a snapshot of oxygen levels, water nutrient levels, chlorophyll-*a* levels and water transparency at the mussel farm and compared these data with the corresponding data from several reference areas, bearing in mind that the status of the system is primarily reflected by the abundance, diversity and species composition of benthic organisms (Kraufvelin *et al*. 2001, 2011, Villnäs *et al*. 2011).

## Material and methods

The mussel farm in Kumlinge has the coordinates 60°12.768´N, 20°45.336´E (Fig. 1) and was set up as a pilot facility in 2010 to test its function for nutrient bioextraction (Engman 2013). The complete farming equipment consists of four 120-m-long and 3-m-deep nets (mesh size of 15 cm) fastened to floating plastic pipes that are kept in place by big buoys and anchors. The distance between each pipe is 10 m (Engman 2013). The average water depth at the farm is 8 m and the area is quite exposed to southern and eastern winds and therefore to water currents, with an average bottom speed of 3–4 cm $s^{-1}$, from SW on the western side and from NW on the eastern side of the mussel farm (T. Engman, pers. comm.). Thus the predominant southwesterly current at the mussel farm forms an eddy which turns the current about 90°, making the areas south and southeast of the farm 'downstream' areas. The water exchange rate is rapid and there is no stratification of the water mass. Detailed data on the water quality (fortnightly sampling) from the area during the summer and autumn of 2010, i.e. the first year of mussel recruitment to the off-bottom farm ropes are described in Mäki (2014).

We sampled the area at the end of August 2012, just before final mussel harvesting (Engman 2013), when the mussel farm had been operating for more than two years. We chose this sampling period as ideal for a snapshot, since any effects, positive or negative, on the faunal assemblage beneath the mussel farm are accumulative. This period also reflected maximum productivity in the water mass (Mäki 2014) and in the benthos (Kraufvelin *et al*. 2011). We

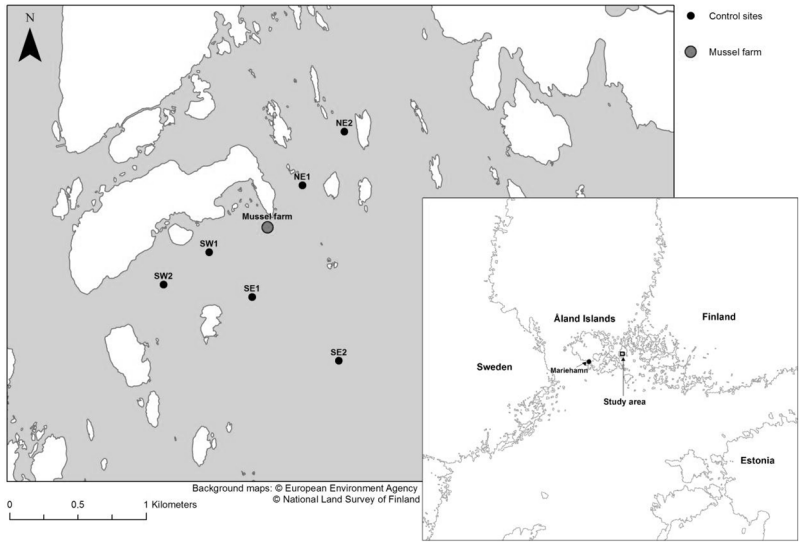


**Fig. 1**. Study area in Kumlinge (Åland Islands): shown are the mussel farm and the reference sites. The major water currents go from SW on the western side of the mussel farm and from NW on the eastern side of the mussel farm. SW, SE and NE in the sample labellings indicate south west, south east and north east, respectively, with regard to directions away from the mussel farm.

examined the organic content (as loss on ignition) and zoobenthos from four sediment grab samples (Petersen grab with inner surface = 250 $cm^2$) from each of the following three stations: (1) beneath the mussel farm, (2) at one reference site upstream, 500 m SW, and (3) at one reference site downstream of the farm, 500 m towards SE. For sediment organic content, we took small (< 1% of the total amount of surface sediment) core samples from the upper 1 cm surface of the sediment and stored these samples in 70% ethanol until analysis. We sieved the rest of the sediment sample for zoobenthos using a 0.5 mm mesh and stored fauna samples in 70% ethanol until we sorted and counted the animals in the laboratory using a dissecting microscope. We identified all organisms to the lowest possible taxonomic level. Additionally, we took snapshot water samples around the farm, increasing the number of reference sites, to complement the faunal analysis. The sampling station at the mussel farm was placed exactly in the middle of the farm area and the reference sites were 500 and 1000 m away from the farm into three directions: SW, SE and NE (Fig. 1). We recorded the Secchi depth and measured water temperature, salinity and oxygen saturation *in situ* electronically. We collected water samples for chlorophyll-*a*, total N and total P with a Limnos sampler and transported them to the laboratory in glass flasks kept in the dark for later analysis. We collected water from 2 and 6 m depths, at each of 7 stations (6 reference sites and the mussel farming site). All stations had a maximum depth of 8 m.

We statistically analysed differences in sediment and water quality (sediment organic content, P concentration, N concentration, chlorophyll *a*) as well as zoobenthos (total abundance, total number of species), using one-way or two-way ANOVA (with *a post hoc* Student–Newman–Keuls (SNK) test where relevant) after checking for normality and homogeneity of variances and using appropriate data transformations when assumptions were violated (for abundance data, $\ln(x + 1)$-transformation was used). We analysed differences in zoobenthos species composition using non-parametric multivariate techniques within the PRIMER statistical package (Clarke 1993), i.e. nMDS ordination, ANOSIM and SIMPER on square-root-transformed data in order to balance the influence between more dominant or rare species/taxa.

## Results

### Sediment and macrofauna analyses

The collected sediments appeared well-aerated on visual inspections during sampling and we observed no differences in oxygen saturation

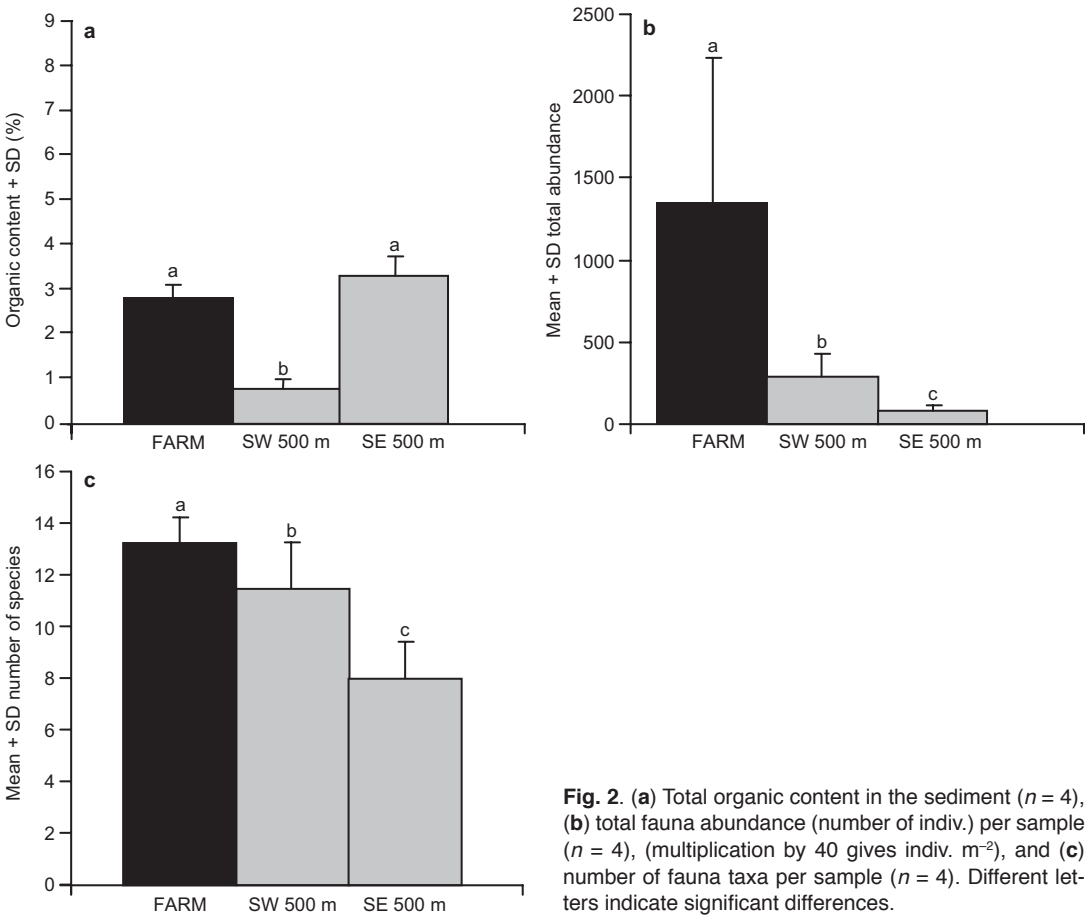


**Fig. 2**. (**a**) Total organic content in the sediment ($n$ = 4), (**b**) total fauna abundance (number of indiv.) per sample ($n$ = 4), (multiplication by 40 gives indiv. $m^{-2}$), and (**c**) number of fauna taxa per sample ($n$ = 4). Different letters indicate significant differences.

at 6–8 m depth (values were always > 90%). Percentage of sediment organic content differed significantly among the three sites (one-way ANOVA: $F_{2,9} = 55.32$, $p < 0.001$); over the farm area (mean ± SD = 2.84% ± 0.30%) and the two reference sites, the upstream site 500 m towards SW (0.78% ± 0.03%) and the downstream site 500 m to the SE (3.32% ± 0.44%). A pair-wise SNK test showed that the organic content was significantly higher within the farm area as compared with that at the reference site 500 m to the SW, but there were no significant differences between the farm area and the reference site 500 m to the SE, downstream (Fig. 2a).

With regard to sediment macrofauna, there were clear differences between the mussel farm site and the two reference sites 500 m to the SW (upstream) and 500 m to the SE (downstream). Total abundance per sample was significantly higher beneath the mussel farm as compared with that at the two reference sites ($F_{2,9} = 18.58$, $p < 0.001$, and SNK test), but there were also significant differences between the reference sites (Fig. 2b). The differences between the mussel farm and the reference sites were especially evident in the large number of blue mussels that apparently had dropped from the ropes (*see* also Engman 2013), but also from macrofauna species associated with these mussel assemblages such as several species of benthic crustaceans and many bioturbators. Hence, the number of taxa was significantly different among sites (one-way ANOVA: $F_{2,9} = 14.49$, $p = 0.002$, and SNK test) revealing a higher number of species in the mussel farm area as compared with that at the reference sites (Fig. 2c) and also more species-rich fauna in the SW (upstream) than in the SE (downstream).

The species/taxa composition (abundance data) of the zoobenthic communities differed

significantly overall (ANOSIM Global $R = 0.95$, $p < 0.001$) and also between all three sites, i.e. the mussel farm and the two reference sites ($R = 1.00$, $p = 0.029$ for all comparisons, which is the lowest possible $p$ value for this permutation test with these numbers of sites and replicates). These differences can be seen in Table 1 and they are also apparent in an nMDS-ordination with Bray-Curtis similarities superimposed, where three distinct groups are formed with 60% similarity (Fig. 3). When trying to reveal the species responsible for these structural differences, it became evident that most species were more numerous beneath the mussel farm (Tables 1 and 2). This was of course true for *M. trossulus*, but also for crustaceans like *Gammarus* sp. (mainly *G. salinus*), *Jaera albifrons*, *Idotea balthica*, Ostracods and *Amphibalanus improvisus*, most of which seemed to be more closely associated with the blue mussels than with the sediment itself. The number of bioturbators and typical background sediment taxa in the area such as *Hydrobia* sp., *M. balthica*, *Marenzelleria neglecta* and Chironomidae was also higher at the mussel farm. The two reference sites only presented higher numbers of individuals for *H. diversicolor* and *M. arenaria* (Tables 1 and 2).

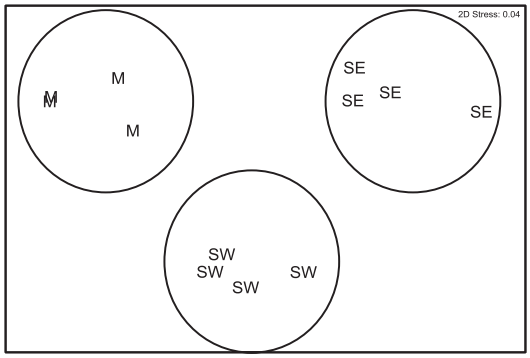


**Fig. 3**. NMDS-ordination (stress 0.04) with 60% Bray-Curtis similarities superimposed summarizing differences in species composition between samples taken beneath the farm (M) and at reference sites 500 m towards SW (upstream) and 500 m towards SE (downstream) of the farm.

## Snapshot of water quality

There were also significant differences in water quality between the mussel farm and the reference sites (Fig. 4). For total P, these differences occurred at 2 m depth ($F_{6,14} = 2.94$, $p = 0.045$) and a pair-wise SNK test demonstrated that they were due to lower values in the farming area (mean ± SD: 17.2 ± 0.9 $\mu$g $l^{-1}$) as com-

**Table 1**. Abundances (mean ± SD, indiv. $m^{-2}$) of the resident sediment macroinvertebrate species/taxa sampled.

| Taxon | Mussel farm | 500 m towards SE | 500 m towards SW |
|---|---|---|---|
| *Prostoma graecense* | 120 ± 142 | – | – |
| Nematoda | – | 80 ± 57 | – |
| *Hediste diversicolor* | – | 220 ± 155 | 70 ± 95 |
| *Marenzelleria neglecta* | 880 ± 497 | 300 ± 168 | 710 ± 390 |
| Sabellidae | – | 70 ± 60 | – |
| Oligochaeta | 40 ± 46 | 210 ± 154 | 10 ± 20 |
| Ostracoda | 1660 ± 1300 | 10 ± 20 | 380 ± 232 |
| *Amphibalanus improvisus* | 290 ± 449 | – | – |
| *Idotea balthica* | 3080 ± 2027 | 620 ± 524 | – |
| *Jaera albifrons* | 2380 ± 2122 | 30 ± 60 | – |
| *Gammarus* spp. | 3380 ± 530 | 30 ± 38 | 10 ± 20 |
| *Calliopius laeviusculus* | – | – | 10 ± 20 |
| Acarina | 20 ± 40 | – | – |
| Chironomidae | 780 ± 421 | 30 ± 38 | 110 ± 88 |
| *Hydrobia* spp. | 11030 ± 6201 | 6920 ± 4536 | 560 ± 364 |
| *Theodoxus fluviatilis* | 20 ± 40 | 280 ± 460 | – |
| *Lymnaea* spp. | 20 ± 40 | – | – |
| *Mytilus trossulus* | 26160 ± 22025 | 210 ± 182 | 20 ± 40 |
| *Cerastoderma glaucum* | 760 ± 499 | 1490 ± 964 | 40 ± 33 |
| *Mya arenaria* | 110 ± 136 | 330 ± 161 | 140 ± 106 |
| *Macoma balthica* | 3360 ± 1151 | 820 ± 364 | 1290 ± 655 |

pared with those at the reference sites upstream and downstream from the farm (20.1–21.3 ± 0.9 $\mu$gl$^{-1}$) (Fig. 4a). For total N, there were no significant differences. The Secchi depths were always greater at the mussel farm (5.1–6.1 m compared with 3.0–4.9 m at reference sites), indicating differences in water transparency (no statistical tests performed). Such differences were also corroborated by chlorophyll-*a* values, where a two-way ANOVA demonstrated a significant interaction between Site and Depth ($F_{6,28}$ = 8.10, $p < 0.001$) and a SNK test revealed that the chlorophyll-*a* levels were clearly lower at 2 m depth at the mussel farm (2.7 ± 0.2 $\mu$g l$^{-1}$) as compared with those at all reference sites, where they ranged between 3.2–3.7 ± 0.1–0.3 $\mu$g l$^{-1}$, except for the site 1000 m to the SE, where the value was 2.8 ± 0.1 $\mu$g l$^{-1}$ (Fig. 4b). At 6 m depth, the chlorophyll-*a* level of 2.7 ± 0.2 $\mu$g l$^{-1}$ was only significantly lower at the mussel farm as compared with that at the SE and NE sites at 1000 m distance (3.1 ± 0.1 and 3.4 ± 0.3 $\mu$gl$^{-1}$, respectively).

## Discussion

Our results from the mussel farm, such as a stimulated macrofauna abundance and species richness as well as lack of hypoxia, departed from the ones predicted by Stadmark and Conley (2011, 2012). It may be explained by the non-commercial size of the mussel farm operating and the influence of local hydrological conditions. The fauna communities beneath the mussel farm (the former fish farm) were visually inspected over time and substrates were repeatedly gauged as already recovered from fish farming by the summer of 2009, determined by normal oxygen levels, lack of hydrogen sulfide smell in the sediments and the presence of a natural fauna (T. Engman pers. comm.). This system therefore seems to be more resilient than predictions suggested in the mussel farm debate. Similarly, we discard the potential for anoxia beneath the mussel farm in this study because the sediments appeared well-aerated on visual inspections associated with sampling. There were no

**Table 2**. SIMPER analyses showing the species responsible for significant differences in species abundance between the mussel farm site and the two reference sites 500 m to the SW (upstream) and 500 m to the SE (downstream) using square-root transformed data and average abundance per sample. The average dissimilarity was 54.89% between M and SW, 66.42% between M and SE and 54.80% between SW and SE.

| Taxon | Average abundance | | | Contribution (%) | | |
|---|---|---|---|---|---|---|
| | Farm | SW | SE | Farm to SW | Farm to SE | SW to SE |
| *Mytilus trossulus* | 23.45 | 1.94 | 0.35 | 26.33 | 27.56 | 4.79 |
| *Gammarus* spp. | 8.79 | 0.60 | 0.25 | 10.32 | 10.52 | 1.65 |
| *Jaera albifrons* | 6.95 | 0.43 | 0 | 7.92 | 8.23 | 1.20 |
| Ostracoda | 6.14 | 0.25 | 2.96 | 7.58 | 3.86 | 7.63 |
| *Hydrobia* spp. | 16.07 | 12.50 | 3.59 | 7.09 | 15.32 | 24.12 |
| *Idotea balthica* | 8.38 | 3.35 | 0 | 6.65 | 10.86 | 9.85 |
| *Macoma balthica* | 9.04 | 4.45 | 5.52 | 5.95 | 4.63 | 4.43 |
| Chironomidae | 4.25 | 0.60 | 1.41 | 4.98 | 3.88 | 3.11 |
| *Cerastoderma glaucum* | 4.15 | 5.81 | 0.85 | 3.22 | 4.04 | 13.62 |
| *Hediste diversicolor* | 0 | 2.02 | 0.91 | 2.75 | 1.15 | 4.67 |
| *Marenzelleria neglecta* | 4.53 | 2.65 | 4.08 | 2.53 | 1.81 | 4.24 |
| *Amphibalanus improvisus* | 2.01 | 0 | 0 | 2.41 | 2.33 | 0 |
| *Mya arenaria* | 1.16 | 2.80 | 1.76 | 2.31 | 1.60 | 3.18 |
| *Theodoxus fluviatilis* | 0.35 | 1.72 | 0 | 2.29 | 0.59 | 4.71 |
| Oligochaeta | 0.71 | 1.97 | 0.25 | 2.19 | 0.95 | 5.49 |
| Nematoda | 0 | 1.22 | 0 | 1.64 | 0 | 3.52 |
| *Prostoma graecense* | 1.22 | 0 | 0 | 1.61 | 1.59 | 0 |
| Sabellidae | 0 | 1.12 | 0 | 1.45 | 0 | 2.97 |
| Acarina | 0.35 | 0 | 0 | 0.40 | 0.38 | 0 |
| *Lymnaea* spp. | 0.35 | 0 | 0 | 0.38 | 0.35 | 0 |
| *Calliopius laeviusculus* | 0 | 0 | 0.25 | 0 | 0.36 | 0.83 |

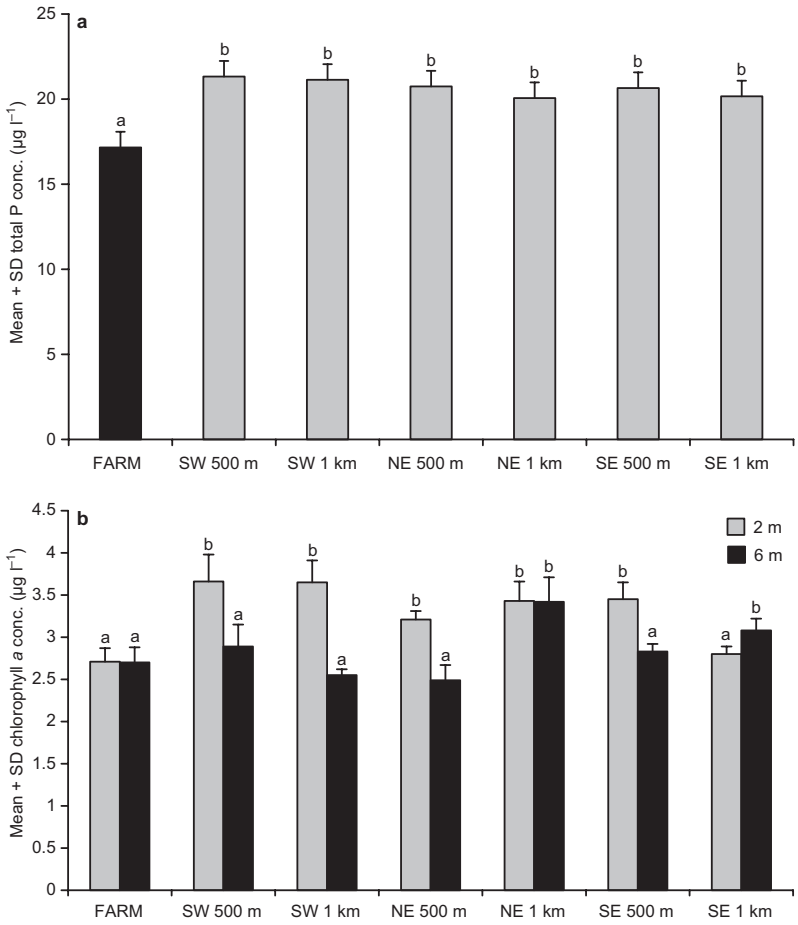


**Fig. 4**. (**a**) Total P concentration at 2 m depth ($n$ = 3), and (**b**) chlorophyll-*a* concentration at 2 and 6 m depths ($n$ = 3). Different letters indicate significant differences.

significant differences in dissolved oxygen saturation at waters of 6–8 m depth (values always > 90%) and the macrofauna appeared to be species rich and abundant. Effective water exchange was probably enabled by the relatively high degree of openness to the sea, together with the shallow water depth and relatively strong bottom currents, where an average current velocity of 3–4 cm $s^{-1}$ may imply maxima of up to 20–30 cm $s^{-1}$ during storms (Lauringson *et al*. 2009, Leino *et al*. 2011). Dominant currents are southwesterly and generally create an eddy 20 m north of the mussel farm and drive the waters south (Engman 2013, Mäki 2014), while simultaneously diluting farm-derived organic matter.

The percentage of sediment organic content differed significantly, being overall higher within the farm area as compared with that at the upstream reference site 500 m towards SW, but there was no statistical difference between the farm area and the downstream reference site 500 m towards SE (Fig. 2a). This was partly in agreement with the expectations by Stadmark and Conley (2012), i.e. a potential concentration of organic matter from a larger area to just one place. It could, however, also partly be a carry-over effect due to the 'historical' fish farming activities (with a yearly production of rainbow trout around 40–50 tonnes), that occurred in the area until 2008 and that may have had long-lasting effects on the organic content both beneath the farm and downstream of it (*see* e.g. Karakassis *et al*. 2000, Kraufvelin *et al*. 2001, Villnäs *et al*. 2011 for similar results). Nevertheless, the highest organic levels in this study, which were in the range of 2.8%–3.3%, were generally lower than values reported for 'unpolluted locations' around the Åland Islands.

For example, Kraufvelin *et al*. (2011) studied three 1-km$^2$ areas and reported background ranges of 0.3%–2.1% for one site; 5.2%–7.5% for another site and 2.9%–10.1% for the third site. At sites formerly affected by fish farms in the northern Baltic proper, the reported ranges were around 8%–12% (Kraufvelin *et al*. 2001) and 5%–20% (Villnäs *et al*. 2011) many years after fish farming abated. The organic levels at this mussel farming site were also clearly lower than values reported for mussel farms globally, which typically ranges from 10%–15% beneath the farms (Stenton-Dozey *et al*. 1999, Chamberlain *et al*. 2001, Ysebaert *et al*. 2009, Ivanov *et al*. 2013, Neofitou *et al*. 2014). It is, however, worth noting that if the mussel farm production increased dramatically, the associated organic matter could over time increase to harmful amounts. Increased production of farmed mussels could also cause an increase in the amounts of mussel clumps falling down to the sediment through intra-specific competition, storm and ice damage or bird predation (Inglis and Gust 2003, Hartstein 2005, Engman 2013). The effects of these clumps on sediment oxygenation should be investigated in future studies. For now, the load from this farm seems to have been quite low or effectively diluted. Sufficient spacing between mussel farming units probably also contributed to the benign conditions.

The mussel farm's high species richness and abundance (up to 8 times the average number of individuals) of sediment macrofauna including many bioturbators, may provide extra resilience to the ecosystem against anoxia and/or enable fast recovery of the benthic communities after mussel harvesting (Norkko and Shumway 2011, Norkko *et al*. 2012). It may also indicate high amounts of organic matter and favourable conditions for bioturbators (rather than crustaceans such as *Monoporeia*), according to the Pearson and Rosenberg (1978) theory. The generally higher values for both number of species and individuals, could also be partly due to the relatively low background eutrophication level in the area, where a moderate additional enrichment may actually stimulate species' densities and richness (Pearson and Rosenberg 1978, Kraufvelin *et al*. 2006a, 2006b, Lauringson and Kotta 2006, Kotta *et al*. 2008, 2009b, Wilding and Nickell 2013). The high values that we found can also partially be explained by the presence of mussel clumps (living individuals or shells) as biodeposits originating from the mussel ropes and falling down onto the sediments beneath the mussel farm. These clumps also serve as shelter/substrate for several macroinvertebrates and are contributing to the distinct species composition beneath the farm and the deviations from the two reference sites (*see* also Wong and O'Shea 2011). Both of which are more similar to other sediment communities around the Åland Islands (e.g. Kraufvelin *et al*. 2011, Villnäs *et al*. 2011). The mussel assemblages are thereby increasing the richness and abundance of species in the sediment communities at the mussel farm site, which can be seen as a positive effect on local biodiversity (Kaiser *et al*. 1998, Norling and Kautsky 2007, 2008, Koivisto *et al*. 2011, Koivisto and Westerbom 2012). Further, they could be attracting fish and birds as well as other components of the aquatic food web. However, it should be stressed that we measured equivalent amounts of sediment in the samples taken beneath the mussel farm and at the reference sites (not only blue mussels with dead understory layers/matrix) and that the macrofauna was typical for the region. Thus, small farms do not seem to be a risk for the benthic environment, instead they would probably bring benefits residing in the extra incorporation of bioturbators which may protect the sediments from hypoxia and anoxia.

This mussel farm in Kumlinge was clearly smaller than facilities used in other marine areas, where the impact has often been related to the total production (e.g. Carlsson *et al*. 2009, 2012, Ivanov *et al*. 2013). The harvestable production of mussels in Kumlinge was around 25 tonnes in 2 years, in comparison with annual local amounts of 100–500 tonnes of mussels in western Sweden and the Danish Fjords, for example. Nevertheless, the mussel settlement onto the farm nets had apparently been efficient at this specific site during the summer of 2010 and this was also supported by results from the study by Mäki (2014). Mussel growth also seemed to have been quite efficient over the 2.5 years with some two year-old mussel individuals even being > 30 mm long (Engman 2013). This growth rate exceeds previously reported values

for blue mussels in the northern Baltic proper at the lower limits of their physiologically tolerable salinity levels (Kautsky 1982, Westerbom *et al*. 2002, Riisgård *et al*. 2014).

The snapshot study on water quality in the area indicated lower P-concentrations and chlorophyll-*a* levels close to the farm and higher water transparency indicating a potential nutrient uptake and effective filter-feeding of plankton by the farmed mussels. A reduction in total P may also decrease blooms of green annual filamentous and sheet-like macroalgae (e.g. Kraufvelin *et al*. 2010) and harmful bluegreen algae (Paerl and Otten 2013). Clear waters in the close proximity to a mussel farm may further stimulate photosynthetic production (via increased depth penetration of light), in the area by generally more 'desirable species', such as indicators of good conditions or unaltered waters, like eelgrass and bladder-wrack (Kautsky *et al*. 1986, Torn *et al*. 2006, Petersen *et al*. 2012, Schröder *et al*. 2014). Their presence could further serve to improve local oxygen conditions. A higher degree of water transparency around this mussel farm as compared with that in the surrounding areas was repeatedly observed in association with plankton blooms during the operative years of the farm (Engman 2013). Finally, the lack of differences in total nitrogen concentration may indicate that the mussel farm is perhaps releasing some nitrogen in the form of ammonium, thereby somewhat counteracting the beneficial nutrient uptake.

The results of this study suggest that the size of the mussel farm is an important aspect to be considered, in combination with the natural hydrological conditions of the location, to enable environmentally sustainable activities. The spatial configuration of the mussel farm can also play an important role in determining the water circulation and thereby food delivery for mussels as well as the potential accumulation of sediments which can be harmful to the ecosystem. Future discussion should thus aim to model and estimate the critical mussel farm production per area before inducing hypoxia or further release of nutrients from the sediment. Concurrently, an increase in background knowledge with regard to the efficiency of mussel farms for combating eutrophication in the Baltic area at larger scales is necessary. Moreover, there is great scope for improvement to future study designs. Since there were no other mussel farming sites available in the Baltic proper and a lack of continuous background data from the study site while the mussel farm was inaccessible (due to final harvesting of mussels during November 2012), our data are inconclusive regarding actual causality issues. In order to deal with causality one would need to apply beyond-BACI-type (BACI = Before-After-Control-Impact) designs (*see* Stewart-Oaten *et al*. 1992, Underwood 1996), as for example was done by Kraufvelin *et al*. (2002) in their controlled whole-ecosystem study on nutrient enrichment. Similarly, an ACI (After-Control-Impact) type of design (e.g. Terlizzi *et al*. 2005) was no alternative, since the mussel harvesting ended the investigation.

The risks of mussel farms altering nutrient biogeochemical cycles and contributing to the general eutrophication of the Baltic Sea (Stadmark and Conley 2011, 2012) seem to be variable over time and space and directly related to the production of the mussel farm; farming methods used and site conditions (Kaiser *et al*. 1998, Miron *et al*. 2005, McKindsey *et al*. 2011, Petersen *et al*. 2012, Rose *et al*. 2012). At this small farming unit, we found no signs of hypoxia and the amount of pseudofeces was relatively small or effectively diluted. If present, however, such negative effects could partly outweigh the expected positive impact of harvesting nutrients via mussel harvest. Due to this, it is crucial to expand this study to mussel farms of different sizes and compare results from different locations in the Baltic Sea, while bearing in mind differences in salinity, which greatly affect the growth and recruitment of mussels (Westerbom *et al*. 2002). This should be done in close connection with examination of the water exchange rate because this is not only a proxy for good recruitment and mussel growth, but is also essential for minimizing the possible negative environmental impacts. Additionally, the need for well-designed environmental monitoring programs running alongside mussel farming activities should be emphasized. These programs should ideally use multiple farming sites; reasonable numbers of reference sites distributed in space; repeated sampling events over time and

the programs should already be initiated before the mussel farms are in place.

To summarize, although the benefits of mussel farming for mitigation of nutrient enrichment are debated, it is likely that mussel farming activities will increase in the Baltic proper in the near future and this study provides important baseline information regarding ideal conditions for setting up mussel farms *a priori*. Even though this contribution was conducted once off at one locality, it was conducted at the maximum density of farmed mussels in the area, so the influence on sediments and on fauna should be reliable (right before final harvesting). Furthermore, this study was conducted at the only mussel farming site in the Baltic proper to date, so there were no real opportunities for spatial/temporal replication/repetition in the short term (Kraufvelin 1999). Finally, it must also be stressed that this was the first time when real data were brought into the mussel farming debate in the Baltic Sea, which until now has been based on arguments, expectations and fears derived from sites outside the Baltic and from the general perception of the critical environmental conditions of the Baltic Sea as a whole. Thus far, our results demonstrates that mussel farms are likely to aid in the local remediation of the eutrophication problem of the Baltic Sea, rather than exacerbate it, in agreement with pioneering papers within the field (e.g. Edebo *et al*. 2000, Lindahl *et al*. 2005, Lindahl and Kollberg 2009).

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